

AperTO - Archivio Istituzionale Open Access dell'Università di Torino

## How to assess the impact of fine sediments on the macroinvertebrate communities of alpine streams? A selection of the best metrics

### This is the author's manuscript

*Original Citation:*

*Availability:*

This version is available <http://hdl.handle.net/2318/1650425> since 2023-01-27T10:35:41Z

*Published version:*

DOI:10.1016/j.ecolind.2017.08.041

*Terms of use:*

Open Access

Anyone can freely access the full text of works made available as "Open Access". Works made available under a Creative Commons license can be used according to the terms and conditions of said license. Use of all other works requires consent of the right holder (author or publisher) if not exempted from copyright protection by the applicable law.

(Article begins on next page)

# How to assess the impact of fine sediments on the macroinvertebrate communities of alpine streams? A selection of the best metrics

Alberto Doretto<sup>a+</sup>, Elena Piano<sup>a+\*</sup>, Francesca Bona<sup>a</sup>, Stefano Fenoglio<sup>b</sup>

<sup>a</sup>DBIOS, University of Turin, Via Accademia Albertina 13, I-10123, Turin, Italy

<sup>b</sup>DISIT, University of Piemonte Orientale, Viale Teresa Michel 25, I-15121, Alessandria, Italy

\*Corresponding author: elena.piano@unito.it

<sup>+</sup>Alberto Doretto and Elena Piano equally contributed to this study

## Highlights

- Siltation results in alterations of the aquatic habitat and biological communities
- To face these sediment-associated impacts specific biomonitoring tools are needed
- Macroinvertebrates are good indicators of physical alterations, including siltation
- We compared the response of several invertebrate community metrics to fine sediment
- We aggregated the most sensitive ones into a stressor-specific multimetric index

29    **Abstract**

30    Excessive fine sediment accumulation (i.e., siltation) in streams and rivers originates from several  
31    human activities and globally results in heavy alterations of aquatic habitats and biological  
32    communities. In this study the correlation between fine sediment and several benthic invertebrate  
33    community metrics was tested through a manipulative approach in alpine streams, where siltation  
34    mainly results as a physical alteration (i.e., the clogging of substrate interstices) without the  
35    influence of co-occurring confounding factors. We selected 12 candidate metrics, belonging to three  
36    different categories: compositional, structural and functional. We first carried out a manipulative  
37    experiment where artificial substrates were used to provide standardized conditions of siltation. All  
38    candidate metrics were calculated for each artificial substrate and the selection of the best  
39    combination of metrics was statistically performed with an information-theoretic approach. All  
40    candidate metrics were calculated both at family level and also at a mixed level (family and genus)  
41    in order to account for the systematic resolution. Then, data from a field study on alpine streams  
42    affected by mining activities were used as independent dataset for testing the performance of the  
43    selected metrics. We found that the total taxa richness, the EPT (Ephemeroptera, Plecoptera and  
44    Trichoptera) richness and the abundance of benthic invertebrates associated to rheophilous  
45    conditions and coarse mineral substrates were the most sensitive metrics. When these metrics were  
46    aggregated into a multimetric index in the validation dataset, we observed high and significant  
47    correlations between index values and the quantity of fine sediment for both taxonomic levels,  
48    especially for the mixed level . The findings of this study provide useful tools for biomonitoring the  
49    effects of fine sediment in low order, mountainous streams and contribute to improve our diagnostic  
50    ability on stressor-specific alterations.

51

52    **Key-words:** siltation, benthic invertebrates, multimetric index, ecological assessment, taxonomic  
53    resolution, rivers

54

55

56

57

58

59

60

## 1. Introduction

The riverbed colmation by fine sediment is one of the world-wide causes of alteration in streams and rivers (Owens et al., 2005; Wilkes et al., 2017). Excessive fine sediment inputs can originate from several anthropogenic sources, including agriculture (Benoy et al., 2012; Burdon et al., 2013), deforestation and clear-cut practices (Couceiro et al., 2010), road construction (Kaller and Hartman, 2004; Cocchiglia et al., 2012), mining activities (Smolders et al., 2003; Pond et al., 2008), damming and river flow regulation (Wood and Armitage, 1999; Crosa et al., 2010).

Fine sediment in running waters can act as a disturbance not only as suspended solids but also as settled material and negative consequences of sedimentation on all the components of lotic ecosystems have been well documented, regardless of the source (Wood and Armitage, 1997; Henley et al., 2000; Jones et al., 2012). Firstly, the deposition of large amount of fine inorganic material on the riverbed causes the loss of substratum heterogeneity and micro-habitats (i.e., spawning habitat for fish and interstitial spaces for invertebrates). A layer of fine sediment also hinders the oxygen and chemical exchanges between the bottom and the water column, producing anoxic or adverse conditions for benthic organisms (i.e., invertebrates and algae). In addition, fine sediment can cause direct damage to the aquatic organisms, clogging their respiration or feeding anatomical structures, producing an abrasive stress and dislodging them from the substrate (Bilotta and Brazier, 2008).

In the last decades, benthic invertebrates have been increasingly used in biomonitoring programs focused on physical alterations in streams, including fine sedimentation (Mebane, 2001; Cover et al., 2008; Kefford et al., 2010). Indeed, macroinvertebrates have a historical tradition as bio-indicators: their use to assess the ecological status of lotic ecosystems started at the beginning of the 20<sup>th</sup> century (Rosenberg and Resh, 1993; Bonada et al., 2006), so that they are currently the most used group in freshwater biomonitoring around the world (Buss et al., 2015).

Recently, interesting stressor-specific biotic indices have been developed describing the structure of macroinvertebrates biological assemblages based on known or hypothesized tolerances of taxa to fine sedimentation (Table 1). For example, the PSI (Proportion of Sediment-sensitive Index), developed in the UK, scores each benthic invertebrate taxon according to its sensitivity or tolerance to fine sediment (Extence et al., 2013). The final index value is then calculated as the proportion of the most sensitive taxa in the sample (i.e., sampling station), adjusted to their range of abundance. The index ranges between 0 and 100, and based on its value five different quality classes are set, varying from completely un-affected by siltation (80-100) to heavy silted (0-20). Similar attempts have been made by Relyea et al. (2000; 2012) and Hubler et al. (2016) in USA. A different approach is proposed by Murphy et al. (2015), who assigned the scores to macroinvertebrate taxa through a multivariate statistical approach, thus overcoming the expert judgment.

Despite their strong biological and statistical bases, these indices present some critical issues. First, they are based on taxonomic identity, thus spatially dependent to the geographical areas where they have been developed. However, the employment of selected community metrics rather than taxon-identity scores may be a good solution to overcome the bio-geographical limits. This aspect introduces a fundamental question: which are the best macroinvertebrate community metrics related to fine sediment conditions? Literature data show that fine sediment affects several characteristics of macroinvertebrate communities, such as diversity, total abundance, relative abundance of functional groups and behavioral patterns (i.e., drift) (Angradi, 1999; Longing et al., 2010; Descloux et al., 2014). For example, reductions in the taxa richness and abundance of

macroinvertebrates have been typically observed when high levels of siltation occur in the substrate or stream-section, especially among the most sensitive taxa (i.e., EPT – Ephemeroptera, Plecoptera and Trichoptera) (Sutherland et al., 2012; Mathers and Wood, 2016). Conversely, some taxa (i.e., Chironomidae, Oligochaeta) could benefit from the environmental conditions provided by fine sediment (Ciesielka, and Bailey, 2001; Cover et al., 2008). Also, trait-based classifications of macroinvertebrate taxa have been recently used to assess the response of macroinvertebrate assemblages to fine sediment conditions, with noteworthy results (Pollard and Yuan, 2010; Conroy et al., 2016; Wilkes et al., 2017). Many studies have demonstrated that specific functional groups of invertebrates are particularly affected by siltation (Rabeni et al., 2005; Longing et al., 2010; Doretto et al., 2016). For example, among the functional feeding groups (FFGs) several authors have observed a concomitant decrease in the abundance of scrapers and filterers along a gradient of fine sediment occurrence (Bo et al., 2007; Sutherland et al., 2012). When considering the biological and ecological traits, large body-sized, univoltine and external-gilled organisms appear especially disadvantaged by fine sediment as well as rheophilous and stony-associated taxa (Buendia et al., 2013; Bona et al., 2016).

A second problem is represented by the spatial extent. According to Larsen et al. (2009), the best spatial extent for directly relating macroinvertebrate communities to fine sedimentation is the patch-scale, since the response at the reach-scale is mediated by other factors, such as land use. However, in most cases, biotic indices were built on the basis of reach-scaled data, thus hindering the real relationship between macroinvertebrate taxa and fine sedimentation (but see Murphy et al., 2015).

Third, in the majority of these indices benthic invertebrates are systematically identified at species level because these methods rely on species-specific sensitivity/tolerance information (Table 1). However, a similar taxonomic resolution is challenging for a routinely biomonitoring and most of the Environmental Agencies adopt a different systematic level, mainly family or genus. Moreover, species-specific data are not often available for some geographical areas or some invertebrate groups.

Table 1. Fine sediment biotic index recently developed with their systematic and geographical applicability details.

Index	Taxonomic resolution	Geographical area(s)	References
PSI (Proportion of Sediment-sensitive Invertebrates)	Family and species	UK	Extence et al., 2013; Glendell et al., 2013; Turley et al., 2014; 2015; 2016
FSBI (Fine Sediment Bioassessment Index)	Genus	USA	Relyea et al., 2000; 2012
BSTI (Biological Sediment Tolerance Index)	OTU (Operational Taxonomic Units: family, genus, species)	Oregon	Hubler et al., 2016
CoFSI <sub>sp</sub> (Combined Fine Sediment Index)	Genus and species	England and Wales	Murphy et al., 2015

134 Fourth, to our knowledge, biotic indices measuring the response of macroinvertebrates to fine  
135 sedimentation reported in the literature mostly concern the augmentation of fine sediment in  
136 streams caused by agriculture (Turley et al., 2014, 2015; Naden et al., 2016). In lowland areas,  
137 agriculture-induced sedimentation usually results as a widespread and chronic disturbance, often  
138 coupled with organic pollution due to pesticides, fertilizers or urbanization. This may act as a  
139 confounding factor on the response of benthic invertebrate assemblages to fine sediment (Turley et  
140 al., 2016). By contrast, farming and human settlements are generally scarce in mountainous areas  
141 due to their pronounced slope and harsh conditions. Nevertheless, fine sedimentation is today  
142 recognized as a primary cause of alteration in alpine streams, originating mainly by acute, localized  
143 or episodic sources, such as logging, mining, cross-river constructions or reservoir flushing (Crosa  
144 et al., 2010; Milisa et al., 2010; Espa et al., 2015; Bona et al., 2016). These lotic environments are  
145 expected to severely suffer the consequences of fine sediment deposition as they are typically  
146 dominated by coarse substrata and erosive features (Allan and Castillo, 2007). However, currently  
147 few studies have been carried out to investigate the specific effects of fine sediment on benthic  
148 macroinvertebrates in alpine streams (but see Espa et al., 2015; Leitner et al., 2015; Doretto et al.,  
149 2017). The aims of this study are: i) to investigate what are the best macroinvertebrate community  
150 metrics responding to fine sediment deposition in alpine streams and ii) to assess how the  
151 taxonomic resolution could affect the relationship between the metrics and fine sediment. In order  
152 to investigate the relationship between macroinvertebrates and fine sedimentation at the proper  
153 scale, we built up an experimental field study in which standardized conditions of fine sediment  
154 were manipulatively determined using artificial substrata (calibration dataset) within one single  
155 alpine reach. We then tested the validity of our index on field-collected data obtained from several  
156 patches nested into different reaches in two alpine streams (validation dataset).  
157 In particular, we aimed at constructing a multimetric index (MMI) following the algorithm  
158 suggested by Schoolmaster et al. (2013). The goal of the algorithm is to produce a maximally  
159 sensitive MMI from a given set of candidate metrics and a measure of human disturbance through  
160 an information theoretic criterion (Anderson and Burnham, 2002) to inform the process.

## 161 **2. Materials and Methods**

162

### 163 **2.1 Calibration dataset**

164 The study was realized in a homogeneous reach of the upper Po, a typical alpine low-order stream  
165 (Paesana, Monviso Natural Park, NW Italy UTM: 360107E, 4949488N; elevation 730 meters a.s.l.)  
166 (Figure 1). To assess the relationship between fine sediments and benthic macroinvertebrate metrics  
167 at the patch scale in alpine environment, we used artificial substrates to create standardized and  
168 replicable sampling units. We placed artificial substrata in a large and uniform reach of the Po  
169 riverbed, according to a random distribution. Each artificial substratum consisted of a parallelepiped  
170 trap built with a metal net (18 cm long, 6 cm wide and 6 cm high, mesh width 0.8 cm, total volume  
171 = 0.65 dm<sup>3</sup>). We constructed 135 traps, with 3 different levels of clogging. Traps were filled with  
172 different proportions of sand (range size 0.5-1 mm) and pea pebbles (average size 14-20 mm) to  
173 provide three different clogging conditions: 45 traps contained 100% pebbles (without sand, i.e. fine  
174 sediment – WFS), 45 traps contained 50% sand and 50% pebbles (medium level of sedimentation -  
175 MED) and 45 traps contained 66% sand and 33% pebbles (clogging condition – CLO). In the  
176 calibration data, we considered sand proportion as proxy of fine sediment amount.

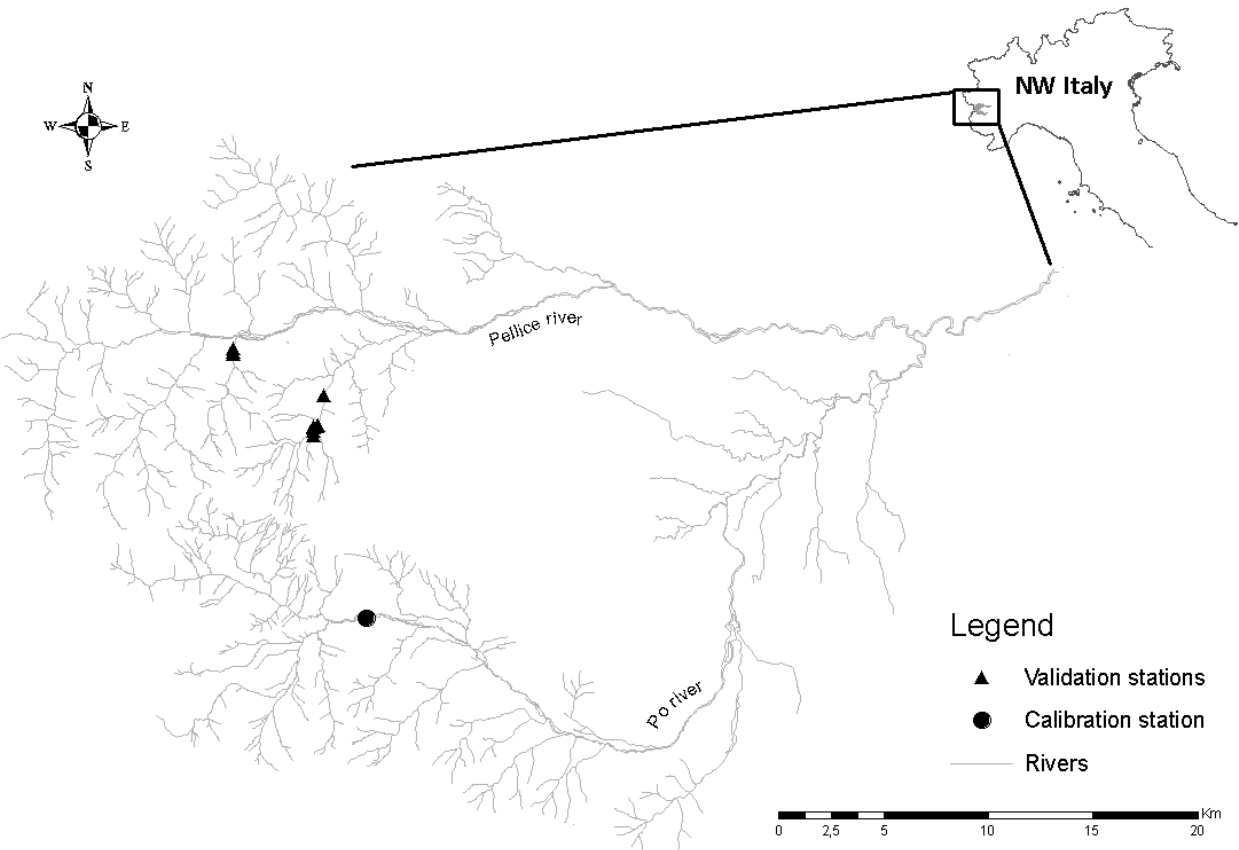
177 All traps were marked with a colored and numbered label and a fine net was applied to their lateral  
178 and basal sides to avoid the loss of fine sediment. Artificial substrata were randomly placed on the  
179 same day, buried in the streambed such that the upper side was flush with the bottom, allowing the  
180 colonization of benthic taxa. We paid attention to guarantee that all artificial substrata were fixed  
181 into the stream bottom with the same orientation and in similar conditions of water depth and  
182 velocity. To evaluate the colonization dynamic of macroinvertebrates on the different clogging  
183 conditions, the artificial substrata were removed on three different sampling dates, namely after 7,  
184 21 and 63 days, for a total of 45 random sampling units (15 for each typology) on each sampling  
185 date. When the cages were removed from the streambed they were suddenly placed into a plastic  
186 bucket and opened. All the content was transferred in separated plastic tins, preserved in 90%  
187 alcohol and returned in laboratory for the sorting and the systematic identification. All benthic  
188 invertebrates were systematically identified until family or genus and counted. Based on their  
189 trophic strategies and their biological and ecological requirements, macroinvertebrates were  
190 classified into the Functional Feeding Groups (FFGs - Merritt et al., 2008) and biological and  
191 ecological traits (Usseglio-Polatera et al., 2000) respectively.

192

## 193 **2.2 Validation dataset**

194 Data for validating the index were collected in a different watershed, comparable to the Po  
195 watershed in terms of physical and chemical variables as well as in terms of human settlement  
196 intensity, to guarantee a wider applicability of the index. For the validation dataset, we thus selected  
197 two third Strahler order streams in the Cottian Alps (Piemonte, NW Italy), the Luserna and the  
198 Comba Liussa streams. They share similar environmental conditions, the only difference being the  
199 presence of quarries in the Luserna which causes augmentation of fine sediments. On the contrary,  
200 the control lotic system is almost unaffected by human activities. Seven reaches were selected  
201 across the Luserna (L1–L7) and three across the Comba Liussa (C1–C3) stream (Figure 1) and in  
202 each of them we selected six roughly equidistant patches. In correspondence of each patch, we  
203 positioned sediment traps, in order to quantitatively characterize each patch in terms of fine  
204 sediment deposition (Bond 2002). Each trap consisted in a plastic storage box ( $165 \times 95 \times 70$  mm),  
205 with a piece of wire mesh ( $20 \times 20$  mm openings; 1.5 mm gauge wire), cut to fit just inside the box  
206 and placed 30 mm from the top of the trap. In the field, the boxes were buried in the streambed such  
207 that their tops were flush with the bottom. Once the boxes were in place, the wire mesh was covered  
208 by a layer of coarse bed material one clast thick. In this way, fine sediments could enter into the  
209 traps, over which local hydraulic conditions were comparable with the whole streambed. All 60  
210 traps were deployed on the same date and removed after 17 days. Samplings were performed when  
211 mining activity in this Alpine area is at its highest level, resulting in an increased load of fine  
212 sediments in the Luserna catchment. This period coincides also with a substantial stability in the  
213 hydrological conditions of the two streams. The fine sediment collected in the traps was returned to  
214 the laboratory, where it was dried and weighted. One benthic sample was collected in each  
215 sampling point, using a Surber sampler (250  $\mu$ m mesh size; 0.062 m<sup>2</sup> area) to evaluate the  
216 macroinvertebrate community. Surber were positioned in the patches of streambed immediately  
217 after the removal of sediment traps and adjacent (laterally) to where traps were placed. Collected  
218 substrate was conserved into plastic jars with 75% ethanol. In the laboratory, all benthic  
219 invertebrates were systematically identified to family or genus as for the calibration dataset and  
220 counted. We then checked if we had representative communities via accumulation curves

221 (Supplementary Material). Based on their trophic strategies and their biological and ecological  
 222 requirements, macroinvertebrates were classified into the Functional Feeding Groups (FFGs -  
 223 Merritt et al., 2008) and biological and ecological traits (Usseglio-Polatera et al., 2000) respectively.



224  
 225 Figure 1. Area of study: circular and triangle dots represent the sampling stations where the  
 226 calibration and validation experiments were respectively carried out.

227  
 228 **2.3 MMI construction**

229 We screened the available literature data in order to detect the macroinvertebrate-based metrics  
 230 most sensitive to fine sediment deposition. Selected potential metrics belonged to the three  
 231 categories indicated by Noss (1990)—*compositional*, *structural* and *functional* metrics—and  
 232 provided ecological information in accordance with the four categories indicated by Hering et al.  
 233 (2006)—*composition/abundance*, *richness/diversity*, *sensitivity/tolerance* and *functional traits*  
 234 (Table 2).

235  
 236 Table 2. Candidate community metrics used in this study and relative categories, ecological  
 237 information and references.

Metric	Category	Ecological information	References
Taxa richness (S)	Compositional	Richness/diversity	Zweig and Rabeni 2001; Buendia et al.



			2013
Ephemeroptera-Plecoptera-Trichoptera richness (EPT S)	Compositional	Richness/diversity	Angradi 1999; Zweig and Rabeni 2001; Pollard and Yuan 2010; Buendia et al. 2013;
Inverse relative abundance of Gasteropoda-Oligochaeta-Diptera (1-GOLD)	Structural	Sensitivity/tolerance	Pinto et al., 2004; Buffagni and Erba 2007
Shannon-Wiener index (H')	Compositional	Richness/diversity	Mebane 2001; Zweig and Rabeni 2001; Buendia et al., 2013
Total abundance (N)	Structural	Composition/abundance	Angradi 1999; Zweig and Rabeni 2001; Buendia et al., 2013;
Ratio between Ephemeroptera,-Plecoptera-Trichoptera and Diptera (EPT/D)	Structural	Composition/abundance	Allan et al., 2006; Aura et al., 2010
Ephemeroptera-Plecoptera-Trichoptera percentage (EPT %)	Compositional	Sensitivity/tolerance	Mebane 2001; Buendia et al., 2013; Conroy et al., 2016
Abundance of Chironomidae	Structural	Composition/abundance	Angradi 1999; Zweig and Rabeni 2001;
Chironomidae/Diptera	Structural	Composition/abundance	Helson and Williams 2013
Shredders/Collector-gatherers	Functional	Functional traits	Merritt et al., 2002; Merritt et al., 2016
Abundance of biological group f (univoltine, large-sized taxa)	Functional	Functional traits	Bo et al., 2007; Bona et al., 2016; Doretto et al., 2017
Abundance of ecological group A (rheophilous and stony-associated taxa)	Functional	Functional traits	Bo et al., 2007; Bona et al., 2016; Doretto et al., 2017

238

239 In addition to the common and widely used taxa richness, Shannon-Wiener index (H') and total  
240 abundance of benthic invertebrates, some metrics referred to key taxonomic groups were also  
241 included. Three of our candidate metrics accounted for the EPT (Ephemeroptera, Plecoptera and  
242 Trichoptera) component: EPT richness, EPT% and the ratio between EPT and Diptera (EPT/D). We  
243 included these metrics because EPT taxa are among aquatic invertebrates the best adapted to  
244 running waters and a key faunal component of the mountain and alpine sections of streams and  
245 rivers (Heiber et al., 2005; Fenoglio et al., 2015). Moreover, they are recognized as the most  
246 sensitive organisms among freshwater invertebrates so that EPT-based metrics are currently  
247 included in biomonitoring indices or programs throughout Europe (Munnè and Pratt, 2009; Gabriels  
248 et al., 2010). Similarly, we focused also on Diptera, Oligochaeta and Gastropoda, resulting in three  
249 different abundance metrics: the abundance of Chironomidae, Chironomidae/Diptera ratio and 1-  
250 GOLD. In general, a strong positive relationship between Diptera, especially Chiromomidae, as

well as Oligochaeta and fine sediment is supported by a huge number of literature data (Smolders et al., 2003; Cover et al., 2008; Descoux, et al., 2013). By contrast, 1-GOLD describes the relative proportion of Gastropoda, Oligochaeta and Diptera in the community/sample. This metric was developed in the European WFD (Water Framework Directive) implementation context and it is currently incorporated in the official Italian biomonitoring index (STAR\_ICMi; Buffagni et al., 2008). The last three community metrics we selected were based on the functional traits of benthic taxa. The shredders/collector-gatherers describes the ratio between invertebrates feeding directly on coarse particulate organic matter (CPOM) and those feeding on fine particulate organic matter (FPOM). In accordance to the River Continuum Concept (Vannote et al., 1980), the former are mainly located in the upper sections of lotic ecosystems (i.e., low-order streams/reaches) as they strongly depend on the allochthonous input of organic matter (i.e., leaves and vegetal detritus) from the riparian areas. By contrast, the abundance of biological group f and ecological group A (*sensu* Usseglio-Polatera et al., 2000) refer to univoltine, large-sized, rheophilous and stony-associated invertebrates respectively. As all these functional metrics encompass taxa associated to the upper sections of streams, characterized by cold, fast-flowing water and large mineral substratum, we decided to include them among the candidate metrics for evaluating the response of macroinvertebrates to fine sedimentation in alpine streams. To evaluate how taxonomic resolution could affect the response of the metrics to the disturbance, all metrics considered were calculated twice: i) the first time they were derived from taxa identified at family level; ii) the second time they were obtained from inventories in which Plecoptera, Ephemeroptera and Turbellaria were identified to genus level in accordance with Italian pre-WFD official biomonitoring tool (I.B.E. - Ghetti, 1997).

In accordance with the data preparation protocol provided by Schoolmaster et al. (2013), before proceeding with the multimetric construction, we removed metrics that contained a large proportion of zero or duplicated another metric and then rescaled them with the formula:

276

277

$$\frac{m - m_{min}}{m_{max} - m_{min}}$$

278

where  $m$  is the observed value of the metric,  $m_{min}$  is the minimum observed value of the metric in the dataset and  $m_{max}$  is the maximum observed value of the metric in the dataset. In this way, all the values ranged from 0 to 1, where 0 corresponds to the worst condition and 1 to the best condition.

Given the manipulative structure of our calibration dataset, we did not adjust metrics for the covariates effects since traps were placed within the same stream reach, being differentiated only by the fine sediment quantity. Metrics positively correlated with fine sediment were reflected and those not correlated were excluded from further analysis.

We then applied the algorithm proposed by Schoolmaster et al. (2013) that can be used to generate a MMI able to discriminate different disturbance conditions from a given set of metrics. This method produces a MMI with the strongest possible negative correlation with human disturbance through statistical inference since it assumes that metrics and the final MMI are linear functions of the measure of disturbance. Potential MMIs are then built as sets of models, where the disturbance represents the dependent variable and metrics are included as independent variables to be tested against the disturbance. In accordance with this protocol, the quantity of fine sediment was chosen as disturbance parameter,  $D$ , which represented the dependent variable in our set of models. In order

to obtain an ordinal distribution for the disturbance parameter in the calibration dataset, traps with 0% of fine sediment were assigned to class 0, traps with 50% of fine sediment were assigned to class 1 and traps with 66% of fine sediment were assigned to class 2. We then applied the algorithm proposed by Schoolmaster et al. (2013). First, we selected an initial metric,  $m_I$  and we added  $m_I$  to each of the rest of the metrics,  $m_j$ , site-by-site; second, for each  $m_j$ , we checked which combination  $m_I + m_j$  had the strongest negative coefficient with  $D$  and we selected that one; third, we added the index to each of the remaining metrics  $m_j$  site-by-site and we selected the combination of index +  $m_j$  that has the strongest negative coefficient with  $D$ ; finally, we continued this process until the log-likelihood ratio test (see Schoolmaster et al., 2013 for further details) reached the threshold of 3.84, which is the value of the chi-squared distribution that corresponds with  $p = 0.05$ . We repeated these steps using all metrics as initial metric and this process resulted in a number of potential MMIs equal to the number of metrics considered. We then compared them according to the AICs and selected the one with the lowest AIC value to choose the best one.

Given the categorical distribution of our disturbance parameter, we performed multinomial linear regressions, specifically conceived for categorical dependent variables, with the function *polr* of the package *MASS* (Venables and Ripley, 2002) in R environment (R Core Team, 2015).

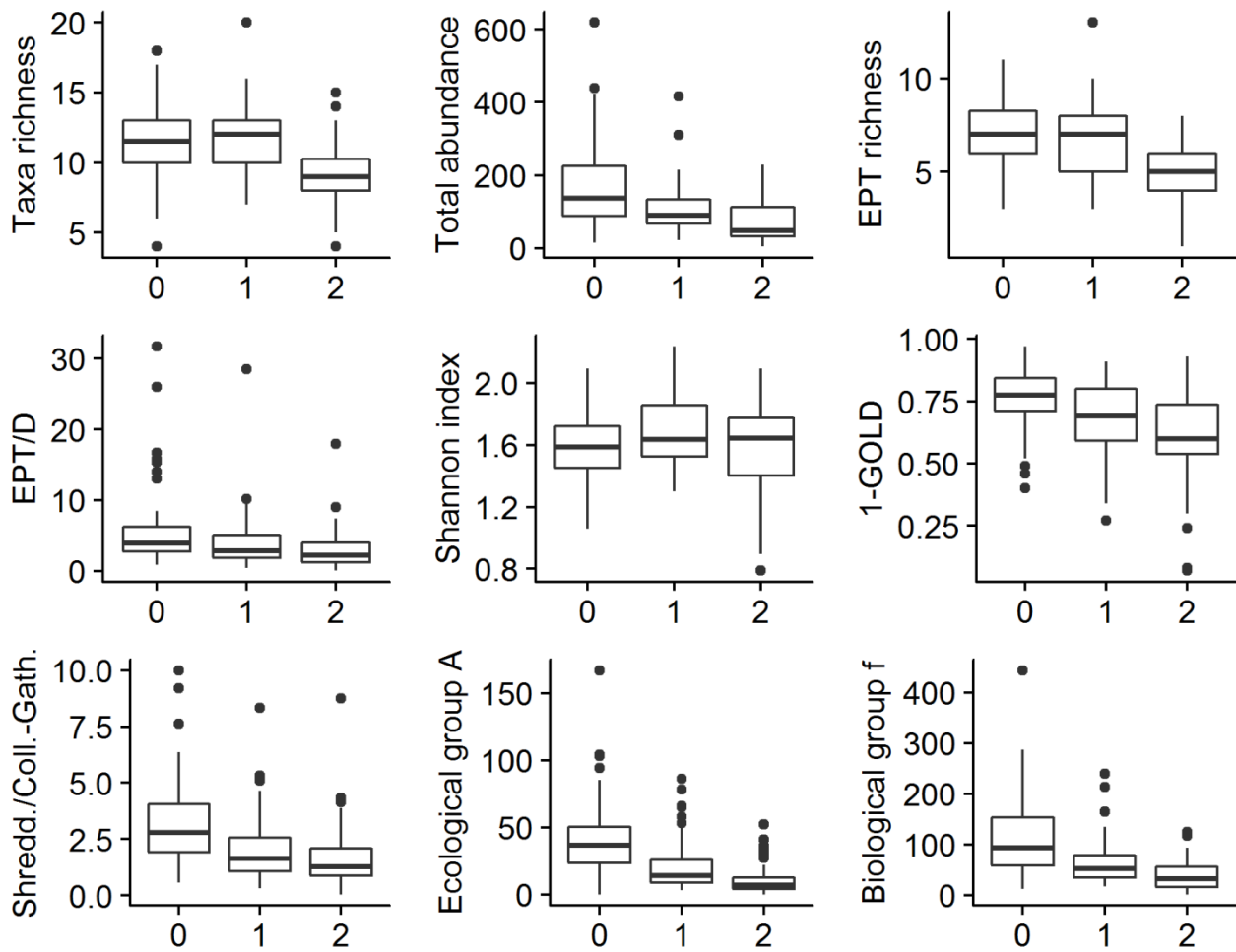
The final MMI was then calculated on the validation dataset by averaging the scaled values (ranging from 0 and 1) of the final selected metrics, obtained from the calibration dataset. The MMI in the validation dataset was calculated for each patch and then averaged for each reach. The field observations of fine sediment were converted into an ordinal variable by calculating the relative proportion of fine sediment weight to the total weight of sediments in each patch and then averaged for each reach. Each observation was then assigned to an ordinal class following the same rules used in the calibration dataset ( $< 50\%$  = class 0;  $50\%$ - $66\%$  = class 1;  $> 66\%$  = class 2). Index values in the validation dataset were comprised from 0 (worst condition) to 1 (best condition) and they were therefore correlated to the fine sediment class for each reach through the Pearson correlation test. This process was repeated for metrics obtained at both mixed (i.e., family and genus) and family levels.

### 3. Results

In the calibration dataset, no relevant morpho-hydrological or chemical changes were observed over the entire sampling period. Overall, the sampling reach was characterized by streambed 7.5-10.3 m wide, cold ( $4.04^{\circ}\text{C} \pm 0.03$  SE), well-oxygenated ( $99.50$  DO%  $\pm 4.45$  SE) and oligotrophic (conductivity:  $172 \mu\text{S}/\text{cm} \pm 0.003$  SE; nitrates =  $0.70$  mg/l; SRP  $< 0.001$  mg/l, BOD<sub>5</sub> =  $3.09$  mg/l) waters. Mean water depth was  $14.4$  cm ( $\pm 0.66$  SE), while the average flow velocity was  $0.07$  m/s ( $\pm 0.003$  SE). We could then exclude an influential effect of environmental parameters on artificial substrata. The average values of the final selected metrics in the three disturbance classes here considered are reported in Supplementary Material.

Before proceeding with the MMI algorithm, we excluded % EPT because in accordance with the protocol provided by Schoolmaster et al. (2013) it can be considered as a duplicate of 1-GOLD. We also excluded the abundance of Chironomidae and the ratio between Chironomidae and Diptera because despite being expected to increase with increasing disturbance they showed an opposite trend. As suggested by Schoolmaster et al. (2013), we excluded them in order to avoid confounding elements. For each of the 9 remnant metrics (Fig. 2), we obtained a potential MMI after the selection process and AIC values are reported in Table 3 and 4.

338  
339



340

341 Figure 2. Boxplots represent the response of the candidate metrics to the sediment conditions in the  
342 calibration dataset: 0 = WFS (0% fine sediment and 100% pebbles), 1 = MED (50% fine sediment  
343 and 50% pebbles), 2 = CLO (66% fine sediment and 33% pebbles).

344

345 Table 3. Final selected models obtained with the family level identification. The AIC column refers  
346 to the AICs values obtained for each model and the  $\Delta$ AIC column refers to the differences between  
347 the AIC of the selected model and the lowest AIC obtained. Values of  $\Delta$ AIC < 2 are reported in  
348 bold.

Potential models	AIC	$\Delta$ AIC
1) Taxa Richness + EPT Richness + Ecological group A	244.42	<b>1.39</b>
2) Total abundance + 1-GOLD + EPT Richness + Taxa Richness	244.70	<b>1.67</b>
3) EPT Richness + Taxa Richness + Ecological group A	244.42	<b>1.39</b>
4) EPT/D + EPT Richness + Taxa Richness + Biological group f	245.38	2.35

5) Shannon + EPT Richness + Taxa Richness + Ecological group A	245.36	2.33
6) 1-GOLD + Total Abundance + EPT Richness + Taxa Richness	244.70	<b>1.67</b>
7) Shredders/Collector-Gatherers + Taxa Richness + EPT Richness + Biological group f	243.03	<b>0.00</b>
8) Ecological group A + Taxa Richness + EPT Richness	244.42	<b>1.39</b>
9) Biological group f + Taxa Richness + EPT Richness + Shredders/Collector-Gatherers	243.03	<b>0.00</b>

349

350 Table 4. Final selected models obtained with the mixed level identification. The AIC column refers  
351 to the AICs values obtained for each model and the  $\Delta AIC$  column refers to the differences between  
352 the AIC of the selected model and the lowest AIC obtained. Values of  $\Delta AIC < 2$  are reported in  
353 bold.

Potential models	AIC	$\Delta AIC$
1) Taxa Richness + EPT Richness + Ecological group A	245.88	<b>1.24</b>
2) Total abundance + 1-GOLD + EPT Richness + Taxa Richness	247.86	3.22
3) EPT Richness + Taxa Richness + Ecological group A	245.88	<b>1.24</b>
4) EPT/D + EPT Richness + Taxa Richness + Biological group f	244.87	<b>0.23</b>
5) Shannon + EPT Richness + Taxa Richness + Ecological group A	247.64	3.00
6) 1-GOLD + Total abundance + Taxa Richness + Ecological group A	244.95	<b>0.31</b>
7) Shredders/Collector-Gatherers + Taxa Richness + EPT Richness + Biological group f	244.64	<b>0.00</b>
8) Ecological group A + Taxa Richness + EPT Richness	245.88	<b>1.24</b>
9) Biological group f + EPT Richness + Taxa Richness	244.98	<b>0.34</b>

354

355 For both the family-level and the mixed-level approach, the MMI assembly algorithm identified 7  
356 out of 9 MMIs with  $\Delta AIC < 2$ . MMIs with values of  $\Delta AIC < 2$  are judged to have substantial  
357 support, and should be considered as viable alternatives to the model with the lowest AIC. In other  
358 words, any MMI can be chosen from the set of models with  $\Delta AIC < 2$  without relevant loss of  
359 predictive power. Starting from this theoretical background, we preferred to select the most  
360 parsimonious solution and we then chose the most recurrent model in both the mixed and family  
361 approaches as the final index, instead of creating weighted indices. In fact, a weighted index,  
362 including all the metrics composing the models with  $\Delta AIC < 2$ , would have been more time-  
363 consuming, because a higher number of metrics should be calculated, without increasing the

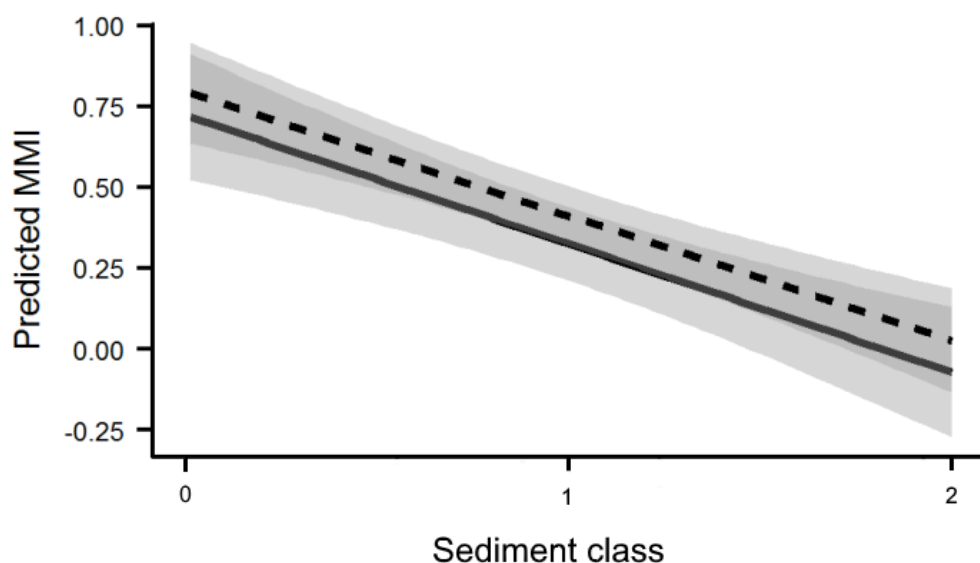
364 predictive power of the index itself. Moreover, it would have required calculating different indices,  
 365 depending on the taxonomic resolution considered, since the final selected models were different  
 366 for the two levels. Our final selected index then was that obtained from equation 1 (Table 3 and 4):

367  $D \sim \text{Taxa Richness} + \text{Ecological group A} + \text{EPT richness}$

368 This model recurred three times for both the mixed and the family level and metrics composing the  
 369 final index also represented the most recurrent ones. In particular, in the family-level procedure,  
 370 Taxa Richness and EPT Richness were included in all 9 alternative models, while Ecological Group  
 371 A was included in 4 out of 9 alternative models. For the mixed-level approach, Taxa Richness was  
 372 included in all the alternative models, EPT Richness was included in 8 out of 9 alternative models,  
 373 the Ecological group A was included in 5 out of 9 alternative models.

374 In the validation dataset, the index was calculated as the average of the scaled values of the final  
 375 metrics. The Pearson correlation test showed high and significant correlations between index values  
 376 and the amount of fine sediment at reach level for both the family-level approach ( $r = -0.73$ ,  $P =$   
 377  $0.017$ ) and the mixed-level approach ( $r = -0.74$ ,  $P = 0.014$ ). The discrimination capacity of the  
 378 index is high for the class 0 of disturbance, while it less powerful in differentiating class 1 and 2,  
 379 especially at the family level (Fig. 3).

380



381

382 Figure 3. Predicted values and confidence intervals of MMIs (continuous line = family level  
 383 approach; dashed line = predicted values derived from the mixed level approach) calculated against  
 384 sediment weight in the validation dataset. Sediment weight is log-transformed for a better graphical  
 385 representation.

386

#### 387 4. Discussion

388 Despite problems associated to fine sediment being widespread, traditional biomonitoring indices  
 389 were developed to detect chemical pollution and usually do not correlate with siltation (Angradi,  
 390 1999; Zweig and Rabení, 2001; Sutherland al., 2012). For this reason, this topic has recently

received an exponential attention so that a new generation of biotic indices has been proposed for measuring the specific effects of fine sediment on macroinvertebrates, mostly in lowland lotic systems (Relyea et al., 2012; Extence et al. 2013; Turley et al., 2014, 2015; Murphy et al., 2015). To our knowledge, our work is the first attempt of building a stressor-specific multimetric index with a manipulative approach aiming at the evaluation of the effect of fine sediment on macroinvertebrates in alpine streams. We are aware of the limitations associated with a manipulative approach, but we are also confident that our experiments resulted highly comparable with real conditions of Alpine streams. In particular, we are confident that the use of sand in the calibration experiment as a proxy of fine sediments could represent a good compromise in Alpine streams. In fact, we had to face some critical issues in the construction, displacement and removal of our traps. It would have been extremely difficult to achieve a similar experimental design using sub-sand fractions, because of high flow velocities and great tractive forces that characterise Alpine running waters. This index may thus represent a promising tool for future biomonitoring assessments for two main reasons: i) using compositional and functional metrics we can overcome the taxonomic constraints intrinsically present in biotic indices (Friberg et al., 2011); ii) this index is highly specific for alpine streams, which represent peculiar ecosystems in which unnatural fine sediment deposition represents one of the main causes of impairment (Wohl, 2006). Compared to other multimetric indices based on macroinvertebrates (Vlek et al., 2004; Couceiro et al., 2012; Mondy et al., 2012), we here introduced an alternative approach, since the calibration was conducted at the patch scale while the validation was performed at the reach scale. A number of other studies have also implied that the ability to detect impacts may be dependent on the choice of sampling scale (Smiley and Dibble, 2008; Burdon et al., 2013). To keep into account the scale of response, we built our index through a manipulative experiment at the patch-scale, which is the most appropriate for measuring the response of macroinvertebrates to fine sediment deposition (Larsen et al., 2009). While field surveys best represent natural conditions, they may be influenced by a range of co-varying factors which may alter biological responses (Matthaei et al., 2010; Robinson et al., 2011; Wagenhoff et al., 2011; Glendell et al., 2014; Turley et al., 2016). Using manipulative experiments allows for the isolation and control of stressors, minimising confounding factors (Kochersberger et al., 2012; O'Callaghan et al., 2015; Piggott et al., 2015; Wang et al. 2016). However, since the stream water quality evaluation and management take place at the reach scale (Collins and Anthony, 2008; Collins et al., 2011; Murphy et al. 2015), it is at this scale that investigations must take place. For this reason, the validity of our index was tested through a second experiment at the reach scale, in order to be applicable for monitoring purposes. In this study, three metrics were retained for their integration into a multimetric index evaluating the impacts of siltation in alpine streams. These were diversity metrics, i.e. total taxa richness and richness in Ephemeroptera, Plecoptera and Trichoptera taxa, and a functional metric, i.e. abundance of rheophilous taxa preferring coarse substrata, typical of oligotrophic, alpine habitats (Ecological group A sensu Usseglio-Polatera et al., 2000). Similar results were obtained by Larsen et al. (2011), who observed a negative effect of fine sediment on both diversity and functional metrics. Our results clearly demonstrated how the combination of different categories of metrics, i.e. diversity and functional metrics, reveal the effect of siltation on biotic communities. This is in accordance with literature (Barbour et al., 1996, 1999; Klemm et al., 2003; Bonada et al. 2006; Hering et al. 2006), since, by combining different categories of metrics, the multimetric assessment is regarded as a more reliable tool than assessment methods based on single metrics. Furthermore, the most

435 relevant combination of appropriate metrics thus consisted of three out of the nine metrics tested. In  
 436 accordance with literature (Menetrey et al. 2011; Schoolmaster et al. 2013), this result shows that  
 437 the selection of the best combination did not include the maximal number of metrics.

438 Concerning diversity metrics, the effect of fine sediment can be significantly measured in terms of  
 439 taxa richness and richness of the most stenoeccious taxa, such as Ephemeroptera, Plecoptera and  
 440 Trichoptera, in agreement with recent studies (Couceiro et al., 2011; Leitner et al., 2015; Conroy et  
 441 al. 2016; Doretto et al., 2017). Although richness metrics are generally sensitive to natural  
 442 variability and seasonality, and thus are influenced by the period of sampling (Bilton al., 2006),  
 443 their combination diminishes this effect and thus increases the robustness of the multimetric index  
 444 (Dahl and Johnson, 2004; Maloney and Feminella, 2006). Given that many biological impacts  
 445 caused by sedimentation are due to sediment deposition (Jones et al., 2012; Glendell et al. 2014),  
 446 we here focused only on deposited fine sediment, excluding suspended sediment.

447 Besides diversity metrics, our findings evidenced that functional metrics may also be effective for  
 448 measuring the impact of fine sediment deposition. Other studies have evidenced changes in trait-  
 449 based metrics to elevated sediment deposition (Rabení et al., 2005; Archaimbault et al., 2010; Bona  
 450 et al. 2016, Turley et al., 2016), generally focusing on functional feeding and habitat groups. In our  
 451 study, the application of ecological and biological groups proposed by Usseglio-Polatera et al.  
 452 (2000), which integrate different aspects, like habitat preferences as well as locomotion, may better  
 453 integrate the filtering effect of siltation on benthic communities (Doretto et al., 2017). The use of  
 454 trait-based indices is a promising approach to help establish the causal relationships between  
 455 specific stressors and macroinvertebrate community response (Doledec and Statzner, 2008; Statzner  
 456 and Beche, 2010, Merritt et al., 2016). In this context, indices and functional trait-based metrics are  
 457 generally considered more sensitive and showed stronger responses to pressures than taxonomy-  
 458 based metrics. Indeed, Dolédec et al. (2006) have demonstrated that functional traits are able to  
 459 integrate more general phenomena than taxonomy-based metrics. Our results clearly support the use  
 460 of functional metrics to build multi-metric indices to assess river biotic integrity.

461 Contrary to our expectations, total abundance of Chironomidae and the Chironomidae/Diptera ratio  
 462 were excluded from the MMI assembly procedure as these metrics showed an inverse relation with  
 463 the disturbance (i.e., amount of fine sediment). In the calibration dataset the highest abundance of  
 464 Chironomidae was detected in the sediment free substrata (WFS), while the lowest in the clogged  
 465 ones (CLO). This finding was unexpected because a huge number of studies have documented high  
 466 densities of Chironomidae midges associated to fine sediment conditions (Ciesielka, and Bailey,  
 467 2001; Kochersberger et al., 2012; Descloux, et al., 2013). However, literature data depict an unclear  
 468 and contrasting situation. Indeed, some authors reported an increment in the Chironomidae  
 469 abundance along a gradient of fine sediment amount, while other authors observed significant and  
 470 opposite responses according to the sub-families of this taxon (Angradi, 1999; Zweig and Rabeni,  
 471 2001). For example, Extence et al. (2013) did not score this family and excluded it for the  
 472 calculation of the PSI because of the wide variability in the sensitivity or tolerance to fine sediment.  
 473 In this study we systematically identified Chironomidae just at family level, losing information on  
 474 the response of each sub-family. This may account for the unexpected response here detected for  
 475 this insect group and it surely represents an important aspect that future studies should consider.

476 Finally, we found that both our family-level and mixed-level (family and genus) MMIs significantly  
 477 correlated with the amount of fine sediment in the validation dataset. However, the discriminant  
 478 capacity was higher for the mixed-level identification than the family level, especially for



disturbance classes 1 and 2. Many authors have demonstrated that a mixed-level systematic identification could improve the performance biotic indices (Schmidt-Kloiber and Nijboer, 2004; Monk et al., 2012). Our findings are in agreement with their results and highlight the importance of the systematic resolution in the freshwater biomonitoring. The choice of the adequate systematic level often reflects a trade-off between the costs associated to the samples processing and the benefits due to species-specific ecological information. Based on our results, we suggest that a mixed-level identification of benthic invertebrates may represent a good solution, with the family as the basic level and the genus for those taxa requiring a higher taxonomic detail, such as EPT or families that encompass a wide range of species. This option may be very advantageous especially for those biomonitoring tools aimed to assess the ecological impairment due to specific stressors.

## 5. Conclusions

We are confident that this study could represent an interesting element in the biomonitoring of siltation impacts. In particular, the index we propose could be effectively employed in alpine environments, considering reach scale and family/genus taxonomic resolution. The fine sediment colmation of riverbed is currently recognized as one of the most widespread forms of alteration by river managers, local agencies and other stakeholders. As a consequence, in the last few years several biotic indices have been developed to specifically quantify the degree of impairment due to anthropogenic fine sediment inputs (Relyea et al., 2012; Extence et al., 2013; Turley et al. 2014, 2015; Murphy et al., 2015; Hubler et al., 2016). All these indices focus on the proportion between sensitive and tolerant benthic invertebrate taxa and rely on valid biological and statistical bases. However, their routine and large-scale applicability appear limited by some aspects, including the systematic resolution and the availability of species-specific data on the sensitivity/tolerance to fine sediment. Moreover, most of them have been developed in an agricultural context and this could represent a confounding factor due to its chemical changes of the water quality. Unlike the above mentioned studies, we tested the correlation between several macroinvertebrate community metrics and fine sediment in alpine streams and we indicated the total richness, the EPT richness and the abundance of rheophilous, stony-associated invertebrates as the best candidate metrics. Other studies have examined the relation between benthic invertebrate community metrics and siltation (Angradi, 1999; Mebane, 2001; Zweig and Rabeni, 2001; Sutherland et al., 2012), but to our knowledge this is the first study aimed to review the candidate metrics and to combine the selected ones into a multimetric index. We are aware that our results need to be validated by further investigations, especially by means of a large-scale survey encompassing a gradient of fine sediment conditions. However, multimetric indices are today widely recommended for biomonitoring purposes as they allow the selection of stressor-specific metrics and the applicability over large geographical areas (Bonada et al, 2006; Nöges et al, 2009; Birk et al., 2012). For example, a multimetric approach was a common consequence of the European Directive 2000/60/EC (Water Framework Directive) and its implementation in many Member States. For these reasons, the findings of this study not only may provide practical tools for biomonitoring the effects of fine sediment but also they may fit with the actual normative scenario.

## Aknowledgements

We are grateful to Elisa Falasco, Anna Chiara Eandi, Davide Giuliano, Sabrina Mossino and Ilaria Zanin for their help in the field work and in laboratory analysis. This work was realized within the

521 framework of Italian MIUR (Ministry of Education, Universities and Research) - PRIN  
522 NOACQUA project.

523

524

525

526

527

528

529

530

531

532

533

534

535

536

## 537 **References**

538 Allan, J.D., Flecker, A.S., Segnini, S., Taphorn, D.C., Sokol, E., Kling, G.W., 2006. Limnology of  
539 Andean piedmont rivers of Venezuela. *J. N. Am. Benthol. Soc.* 25(1), 66–81.

540 Allan, J.D., Castillo, M.M., 2007. Stream ecology. Structure and function of running waters.  
541 Springer, New York.

542 Anderson, D.R., Burnham, K.P., 2002. Avoiding pitfalls when using information-theoretic  
543 methods. *J. Wildlife Manage.* 66 (3), 912–918.

544 Angradi, T.R., 1999. Fine sediment and macroinvertebrate assemblages in Appalachian streams: a  
545 field experiment with biomonitoring applications. *J. N. Am. Benthol. Soc.* 18(1), 49–66.

546 Archaimbault, V., Usseglio-Polatera, P., Garric, J., Wasson, J.G., Babut, M., 2010. Assessing  
547 pollution of toxic sediment in streams using bio-ecological traits of benthic macroinvertebrates.  
548 *Freshwater Biol.* 55(7), 1430–1446.

549 Aura, C.M., Raburu, P.O., Herrmann, J., 2010. A preliminary macroinvertebrate Index of Biotic  
550 Integrity for bioassessment of the Kipkaren and Sosiani Rivers, Nzoia River basin, Kenya. *Lake*  
551 *Reserv. Res. Manage* 15(2), 119–128.

552 Barbour, M.T., Gerritsen, J., Griffith, G.E., Frydenborg, R., McCarron, E., White, J.S., Bastian,  
553 M.L., 1996. A framework for biological criteria for Florida streams using benthic  
554 macroinvertebrates. *J. N. Am. Benthol. Soc.* 15(2), 185–211.

555 Barbour, M.T., Gerritsen, J., Snyder, B.D., Stribling, J.B., 1999. Rapid bioassessment protocols for  
556 use in streams and Wadeable rivers. USEPA, Washington.

557 Benoy, G.A., Sutherland, A.B., Culp, J.M., Brua, R.B., 2012. Physical and ecological thresholds for  
558 deposited sediments in streams in agricultural landscapes. *J. Environ. Qual.* 41(1), 31–40.

559 Bilotta, G.S., Brazier, R.E., 2008. Understanding the influence of suspended solids on water quality  
560 and aquatic biota. *Water Res.* 42(12), 2849–2861.

561 Bilton, D.T., Mcabendroth, L., Bedford, A., Ramsay, P.M., 2006. How wide to cast the net? Cross-  
562 taxon congruence of species richness, community similarity and indicator taxa in ponds. *Freshwater*  
563 *Biol.* 51(3), 578–590.

564 Birk, S., Bonne, W., Borja, A., Brucet, S., Courrat, A., Poikane, S., Solimini, A., van de Bund, W.,  
565 Zampoukas, N., Hering, D., 2012. Three hundred ways to assess Europe's surface waters: an almost  
566 complete overview of biological methods to implement the Water Framework Directive. *Ecol.*  
567 *Indic.* 18, 31–41.

568 Bo, T., Fenoglio, S., Malacarne, G., Pessino, M., Sgariboldi, F., 2007. Effects of clogging on stream  
569 macroinvertebrates: an experimental approach. *Limnologia*, 37(2), 186–192.

570 Bona, F., Doretto, A., Falasco, E., La Morgia, V., Piano, E., Ajassa, R., Fenoglio, S., 2016.  
571 Increased sediment loads in alpine streams: an integrated field study. *River Res. Appl.* 32, 1316–  
572 1326.

573 Bonada, N., Prat, N., Resh, V.H., Statzner, B., 2006. Developments in aquatic insect biomonitoring:  
574 a comparative analysis of recent approaches. *Annu. Rev. Entomol.* 51, 495–523.

575 Bond, N.R., 2002. A simple device for estimating rates of fine sediment transport along the bed of  
576 shallow streams. *Hydrobiologia* 468(1), 155–161.

577 Buendia, C., Gibbins, C.N., Vericat, D., Batalla, R.J., Douglas, A., 2013. Detecting the structural  
578 and functional impacts of fine sediment on stream invertebrates. *Ecol. Indic.* 25, 184–196.

579 Buffagni, A., Erba, S., 2007. Intercalibrazione e classificazione di qualità ecologica dei fiumi per la  
580 2000/60/EC (WFD): l'indice STAR\_ICMi. *IRSA-CNR Notiziario dei metodi analitici* 1, 94–100.

581 Buffagni, A., Erba, S., Pagnotta, R., 2008. Definizione dello stato ecologico dei fiumi sulla base dei  
582 macroinvertebrati bentonici per la 2000/60/EC (WFD): il sistema di classificazione MacOper.  
583 *IRSA-CNR Notiziario dei metodi analitici* 1, 25–41.

584 Burdon, F.J., McIntosh, A.R., Harding, J.S., 2013. Habitat loss drives threshold response of benthic  
585 invertebrate communities to deposited sediment in agricultural streams. *Ecol. Appl.* 23(5), 1036–  
586 1047.

587 Burnham, K.P., Anderson, D., 2002. Model selection and multi-model inference. A Practical  
588 information-theoretic approach. Springer, New York.

589 Buss, D.F., Carlisle, D.M., Chon, T.S., Culp, J., Harding, J.S., Keizer-Vlek, H.E., Robinson, W.A.,  
590 Strachan, S., Thirion, C., Hughes, R.M., 2015. Stream biomonitoring using macroinvertebrates  
591 around the globe: a comparison of large-scale programs. *Environ. Monit. Assess.* 187(1), 4132.

592 Ciesielka, I.K., Bailey, R.C., 2001. Scale-specific effects of sediment burial on benthic  
593 macroinvertebrate communities. *J. Freshwater Ecol.* 16(1), 73–81.

594 Cocchiglia, L., Purcell, P.J., Kelly-Quinn, M., 2012. A critical review of the effects of motorway  
595 river-crossing construction on the aquatic environment. *Freshw. Rev.* 5, 141–168.

596 Collins, A.L., Anthony, S.G., 2008. Assessing the likelihood of catchments across England and  
597 Wales meeting ‘good ecological status’ due to sediment contributions from agricultural  
598 sources. *Environ. Sci. Policy* 11(2), 163–170.

599 Collins, A.L., Naden, P.S., Sear, D.A., Jones, J.I., Foster, I.D., Morrow, K., 2011. Sediment targets  
600 for informing river catchment management: international experience and prospects. *Hydrol.*  
601 *Process* 25(13), 2112–2129.

602 Conroy, E., Turner, J.N., Rymaszewicz, A., Bruen, M., O’Sullivan, J.J., Lawler, D.M., Lally, H.,  
603 Kelly-Quinn, M., 2016. Evaluating the relationship between biotic and sediment metrics using  
604 mesocosms and field studies. *Sci. Tot. Environ.* 568, 1092–1101.

605 Couceiro, S.R.M., Hamada, N., Forsberg, B.R., Padovesi-Fonseca, C., 2010. Effects of  
606 anthropogenic silt on aquatic macroinvertebrates and abiotic variables in streams in the Brazilian  
607 Amazon. *J. Soils Sediments* 10(1), 89–103.

608 Couceiro, S.R.M., Hamada, N., Forsberg, B.R., Padovesi-Fonseca, C., 2011. Trophic structure of  
609 macroinvertebrates in Amazonian streams impacted by anthropogenic siltation. *Austral Ecol.* 36(6),  
610 628–637.

611 Couceiro, S.R.M., Hamada, N., Forsberg, B.R., Pimentel, T.P., Luz, S.L.B., 2012. A  
612 macroinvertebrate multimetric index to evaluate the biological condition of streams in the Central  
613 Amazon region of Brazil. *Ecol. Indic.* 18, 118–125.

614 Cover, M.R., May, C.L., Dietrich, W.E., Resh, V.H., 2008. Quantitative linkages among sediment  
615 supply, streambed fine sediment, and benthic macroinvertebrates in northern California streams. *J.*  
616 *N. Am. Benthol. Soc.* 27(1), 135–149.

617 Crosa, G., Castelli, E., Gentili, G., Espa, P., 2010. Effects of suspended sediments from reservoir  
618 flushing on fish and macroinvertebrates in an alpine stream. *Aquat. Sci.* 72(1), 85–95.

619 Dahl, J., Johnson, R.K., 2004. A multimetric macroinvertebrate index for detecting organic  
620 pollution of streams in southern Sweden. *Arch. Hydrobiol.* 160(4), 487–513.

621 Descoux, S., Datry, T., Marmonier, P., 2013. Benthic and hyporheic invertebrate assemblages  
622 along a gradient of increasing streambed colmation by fine sediment. *Aquat. Sci.* 75(4), 493–507.

623 Descloux, S., Datry, T., Usseglio-Polatera, P., 2014. Trait-based structure of invertebrates along a  
624 gradient of sediment colmation: benthos versus hyporheos responses. *Sci. Tot. Environ.* 466, 265–  
625 276.

626 Dolédec, S., Phillips, N., Scarsbrook, M., Riley, R.H., Townsend, C.R., 2006. Comparison of  
627 structural and functional approaches to determining landuse effects on grassland stream invertebrate  
628 communities. *J. N. Am. Benthol. Soc.* 25(1), 44–60.

629 Doledec, S., Statzner, B., 2008. Invertebrate traits for the biomonitoring of large European rivers:  
630 an assessment of specific types of human impact. *Freshwater Biol.* 53(3), 617–634.

631 Doretto, A., Bona, F., Falasco, E., Piano, E., Tizzani, P., Fenoglio, S., 2016. Fine sedimentation  
632 affects CPOM availability and shredder abundance in Alpine streams. *J. Freshwater Ecol.* 31(2),  
633 299–302.

634 Doretto, A., Bona, F., Piano, E., Zanin, I., Eandi, A.C., Fenoglio, S., 2017. Trophic availability  
635 buffers the detrimental effects of clogging in an alpine stream. *Sci. Tot. Environ.*  
636 <http://dx.doi.org/10.1016/j.scitotenv.2017.03.108>.

637 Espa, P., Crosa, G., Gentili, G., Quadroni, S., Petts, G., 2015. Downstream ecological impacts of  
638 controlled sediment flushing in an Alpine valley river: a case study. *River Res. Applic.* 31(8), 931–  
639 942.

640 Extence, C.A., Chadd, R.P., England, J., Dunbar, M.J., Wood, P.J., Taylor, E.D., 2013. The  
641 assessment of fine sediment accumulation in rivers using macro-invertebrate community response.  
642 *River Res. Applic.* 29, 17–55.

643 Fenoglio, S., Bo, T., Cammarata, M., López-Rodríguez, M.J., Tierno De Figueroa, J.M., 2015.  
644 Seasonal variation of allochthonous and autochthonous energy inputs in an Alpine stream. *J.*  
645 *Limnol.* 74, 272–277.

646 Friberg, N., Bonada, N., Bradley, D.C., Dunbar, M.J., Edwards, F.K., Grey, J., Hayes, R.B.,  
647 Hildrew, A.G., Lamouroux, N., Trimmer, M., Woodward, G., 2011. Biomonitoring of human  
648 impacts in freshwater ecosystems: the good, the bad and the ugly. *Adv. Ecol. Res.* 44, 1–68.

649 Gabriels, W., Lock, K., De Pauw, N., Goethals, P.L., 2010. Multimetric Macroinvertebrate Index  
650 Flanders (MMIF) for biological assessment of rivers and lakes in Flanders (Belgium). *Limnologica*  
651 40(3), 199–207.

652 Ghetti, P.F., 1997. Manuale di applicazione Indice Biotico Esteso (I.B.E.). I macroinvertebrati nel  
653 controllo della qualità degli ambienti di acque correnti. Provincia Autonoma di Trento, Trento.

654 Glendell, M., Extence, C., Chadd, R., Brazier, R.E., 2014. Testing the pressure-specific invertebrate  
655 index (PSI) as a tool for determining ecologically relevant targets for reducing sedimentation in  
656 streams. *Freshwater Biol.* 59(2), 353–367.

657 Helson, J.E., Williams, D.D., 2013. Development of a macroinvertebrate multimetric index for the  
658 assessment of low-land streams in the neotropics. *Ecol. Indic.* 29, 167–178.

659 Henley, W.F., Patterson, M.A., Neves, R.J., Lemly, A.D., 2000. Effects of sedimentation and  
660 turbidity on lotic food webs: a concise review for natural resource managers. *Rev. Fish. Sci.* 8(2),  
661 125–139.

662 Hering, D., Johnson, R.K., Kramm, S., Schmutz, S., Szoszkiewicz, K., Verdonschot, P.F., 2006.  
663 Assessment of European streams with diatoms, macrophytes, macroinvertebrates and fish: a  
664 comparative metric-based analysis of organism response to stress. *Freshwater Biol.* 51(9), 1757-  
665 1785.

666 Hieber, M., Robinson, C.T., Uehlinger, U.R.S., Ward, J.V., 2005. A comparison of benthic  
667 macroinvertebrate assemblages among different types of alpine streams. *Freshwater Biol.* 50(12),  
668 2087–2100.

669 Hubler, S., Huff, D.D., Edwards, P., Pan, Y., 2016. The Biological Sediment Tolerance Index:  
670 Assessing fine sediments conditions in Oregon streams using macroinvertebrates. *Ecol. Indic.* 67,  
671 132–145.

672 Jones, J.I., Murphy, J.F., Collins, A.L., Sear, D.A., Naden, P.S., Armitage, P.D., 2012. The impact  
673 of fine sediment on macro-invertebrates. *River Res. Appl.* 28(8), 1055–1071.

674 Kaller, M.D., Hartman, K.J., 2004. Evidence of a threshold level of fine sediment accumulation for  
675 altering benthic macroinvertebrate communities. *Hydrobiologia* 518(1), 95–104.

676 Kefford, B.J., Zalizniak, L., Dunlop, J.E., Nuggeoda, D., Choy, S.C., 2010. How are  
677 macroinvertebrates of slow flowing lotic systems directly affected by suspended and deposited  
678 sediments? *Environ. Pollut.* 158(2), 543–550.

679 Klemm, D.J., Blocksom, K.A., Fulk, F.A., Herlihy, A.T., Hughes, R.M., Kaufmann, P.R., Peck,  
680 D.V., Stoddard, J.L., Thoeny, W.T., Griffith, M.B., Davis, W.S., 2003. Development and evaluation  
681 of a macroinvertebrate biotic integrity index (MBII) for regionally assessing Mid-Atlantic  
682 Highlands streams. *Environ. Manage.* 31(5), 656–669.

683 Kochersberger, J.P., Burton, G.A., Custer, K.W., 2012. Short-term macroinvertebrate recruitment  
684 and sediment accumulation: A novel field chamber approach. *Environ. Toxicol. Chem.* 31(5),  
685 1098–1106.

686 Larsen, S., Vaughan, I.P., Ormerod, S.J., 2009. Scale-dependent effects of fine sediments on  
687 temperate headwater invertebrates. *Freshwater Biol.* 54(1), 203– 219.

688 Larsen, S., Pace, G., Ormerod, S.J., 2011. Experimental effects of sediment deposition on the  
689 structure and function of macroinvertebrate assemblages in temperate streams. *River Res.*  
690 *Applic.* 27(2), 257–267.

691 Leitner, P., Hauer, C., Ofenböck, T., Pletterbauer, F., Schmidt-Kloiber, A., Graf, W., 2015. Fine  
692 sediment deposition affects biodiversity and density of benthic macroinvertebrates: A case study in  
693 the freshwater pearl mussel river Waldaist (Upper Austria). *Limnologica* 50, 54–57.

694 Longing, S.D., Voshell, J.R., Dolloff, C.A., Roghair, C.N., 2010. Relationships of sedimentation  
695 and benthic macroinvertebrate assemblages in headwater streams using systematic longitudinal  
696 sampling at the reach scale. *Environ. Monit. Assess.*, 161(1), 517–530.

697 Maloney, K.O., Feminella, J.W., 2006. Evaluation of single-and multi-metric benthic  
698 macroinvertebrate indicators of catchment disturbance over time at the Fort Benning Military  
699 Installation, Georgia, USA. *Ecol. Indic.* 6(3), 469–484.

700 Mathers, K.L., Wood, P.J., 2016. Fine sediment deposition and interstitial flow effects on  
701 macroinvertebrate community composition within riffle heads and tails. *Hydrobiologia* 776(1),  
702 147–160.

703 Matthaei, C.D., Piggott, J.J., Townsend, C.R., 2010. Multiple stressors in agricultural streams:  
704 interactions among sediment addition, nutrient enrichment and water abstraction. *J. Appl. Ecol.*  
705 47(3), 639–649.

706 Mebane, C.A., 2001. Testing bioassessment metrics: macroinvertebrate, sculpin, and salmonid  
707 responses to stream habitat, sediment, and metals. *Environ. Monit. Assess.* 67(3), 293–322.

708 Menetrey, N., Oertli, B., Lachavanne, J.B., 2011. The CIEPT: a macroinvertebrate-based  
709 multimetric index for assessing the ecological quality of Swiss lowland ponds. *Ecol. Indic.* 11(2),  
710 590–600.

711 Merritt, R.W., Cummins, K.W., Berg, M.B., Novak, J.A., Higgins, M.J., Wessell, K.J., Lessard,  
712 J.L., 2002. Development and application of a macroinvertebrate functional-group approach in the  
713 bioassessment of remnant river oxbows in southwest Florida. *J. N. Am. Benthol. Soc.* 21(2), 290–  
714 310.

715 Merritt, R.W., Cummins K.W., Berg, M.B., 2008. An introduction to the aquatic insects of North  
716 America. Kendall/Hunt, Dubuque, Iowa, USA.

717 Merritt, R.W., Fenoglio, S., Cummins, K.W., 2016. Promoting a functional macroinvertebrate  
718 approach in the biomonitoring of Italian lotic systems. *J. Limnol.* DOI: 10.4081/jlimnol.2016.1502.

719 Milisa, M., Zivkovic, V., Matonickin Kepcija, R., Habdija, I., 2010. Siltation disturbance in a  
720 mountain stream: aspect of functional composition of the benthic community. *Period. Biol.* 112(2),  
721 173–178.

722 Mondy, C.P., Villeneuve, B., Archaimbault, V., Usseglio-Polatera, P., 2012. A new  
723 macroinvertebrate-based multimetric index (I2M2) to evaluate ecological quality of French  
724 wadeable streams fulfilling the WFD demands: a taxonomical and trait approach. *Ecol. Indic.* 18,  
725 452–467.

726 Monk, W.A., Wood, P.J., Hannah, D.M., Extence, C.A., Chadd, R.P., Dunbar, M.J., 2012. How  
727 does macroinvertebrate taxonomic resolution influence ecohydrological relationships in riverine  
728 ecosystems. *Ecohydrology* 5(1), 36–45.

729 Munné, A., Prat, N., 2009. Use of macroinvertebrate-based multimetric indices for water quality  
730 evaluation in Spanish Mediterranean rivers: an intercalibration approach with the IBMWP  
731 index. *Hydrobiologia* 628(1), 203–225.

732 Murphy, J.F., Jones, J.I., Pretty, J.L., Duerdoth, C.P., Hawczak, A., Arnold, A., Blackburn, J.H.,  
733 Naden, P.S., Old, G.H., Sear, D.A., Hornby, D., Clarke, R.T., Collins, A.L., 2015. Development of  
734 a biotic index using stream macroinvertebrates to assess stress from deposited fine  
735 sediment. *Freshwater Biology*, 60(10), 2019–2036.

736 Naden, P.S., Murphy, J.F., Old, G.H., Newman, J., Scarlett, P., Harman, M., Duerdoth, C.P.,  
737 Hawczak, A., Pretty, J.L., Arnold, A., Laizé, C., Hornby, D.D., Collins, A.L., Sear, D.A., Jones,  
738 J.I., 2016. Understanding the controls on deposited fine sediment in the streams of agricultural  
739 catchments. *Sci. Tot. Environ.* 547, 366–381.

740 Nöges, P., van de Bund, W., Cardoso, A.C., Solimini, A.G., Heiskanen, A.S., 2009. Assessment of  
741 the ecological status of European surface waters: a work in progress. *Hydrobiologia* 633, 197–211.

742 Noss, R.F., 1990. Indicators for monitoring biodiversity: a hierarchical approach. *Conserv. Biol.*  
743 4(4), 355–364.

744 O’Callaghan, P., Jocqué, M., Kelly-Quinn, M., 2015. Nutrient-and sediment-induced  
745 macroinvertebrate drift in Honduran cloud forest streams. *Hydrobiologia* 758(1), 75–86.

746 Owens, P.N., Batalla, R.J., Collins, A.J., Gomez, B., Hicks, D.M., Horowitz, A.J., Kondolf, G.M.,  
747 Marden, M., Page, M.J., Peacock, D.H., Petticrew, E.L., Salomons, W., Trustrum, N.A., 2005.  
748 Fine-grained sediment in river systems: environmental significance and management issues. *River*  
749 *Res. Applic.* 21(7), 693–717.

750 Piggott, J.J., Townsend, C.R., Matthaei, C.D., 2015. Climate warming and agricultural stressors  
751 interact to determine stream macroinvertebrate community dynamics. *Glob. Change Biol.* 21(5),  
752 1887–1906.

753 Pinto, P., Rosado, J., Morais, M., Antunes, I., 2004. Assessment methodology for southern siliceous  
754 basins in Portugal. *Hydrobiologia* 516(1), 191–214.

755 Pollard, A.I., Yuan, L.L., 2010. Assessing the consistency of response metrics of the invertebrate  
756 benthos: a comparison of trait-and identity-based measures. *Freshwater Biol.* 55(7), 1420–1429.

757 Pond, G.J., Passmore, M.E., Borsuk, F.A., Reynolds, L., Rose, C.J., 2008. Downstream effects of  
758 mountaintop coal mining: comparing biological conditions using family-and genus-level  
759 macroinvertebrate bioassessment tools. *J. N. Am. Benthol. Soc.* 27(3), 717–737.

760 R Development Core Team, 2015. R: a language and environment for statistical computing. R  
761 Foundation for Statistical Computing, Vienna, Austria. <http://www.r-project.org>.

762 Rabení, C.F., Doisy, K.E., Zweig, L.D., 2005. Stream invertebrate community functional responses  
763 to deposited sediment. *Aquat. Sci.* 67(4), 395–402.



764 Relyea, C.D., Minshall, G.W., Danehy, R.J., 2000. Stream insects as bioindicators of fine  
765 sediment. *Proceedings of the Water Environment Federation*, 663–686.

766 Relyea, C.D., Minshall, G.W., Danehy, R.J., 2012. Development and validation of an aquatic fine  
767 sediment biotic index. *Environ. Manage.* 49(1), 242–252.

768 Robinson, C.T., Blaser, S., Jolidon, C., Shama, L.N.S., 2011. Scales of patchiness in the response of  
769 lotic macroinvertebrates to disturbance in a regulated river. *J. N. Am. Benthol. Soc.* 30(2), 374–385.

770 Rosenberg, D.M., Resh, V.H., 1993. *Freshwater biomonitoring and benthic macroinvertebrates*.  
771 Chapman & Hall, New York, USA.

772 Schmidt-Kloiber, A., Nijboer, R.C., 2004. The effect of taxonomic resolution on the assessment of  
773 ecological water quality classes. *Hydrobiologia* 516, 269–283.

774 Schoolmaster, D.R., Grace, J.B., Schweiger, E.W., Guntenspergen, G.R., Mitchell, B.R., Miller,  
775 K.M., Little, A.M., 2013. An algorithmic and information-theoretic approach to multimetric index  
776 construction. *Ecol. Indic.* 26, 14–23.

777 Smiley, P.C., Dibble, E.D., 2008. Influence of spatial resolution on assessing channelization  
778 impacts on fish and macroinvertebrate communities in a warmwater stream in the southeastern  
779 United States. *Environ. Monit. Assess.* 138(1), 17–29.

780 Smolders, A.J.P., Lock, R.A.C., Van der Velde, G., Medina Hoyos, R.I., Roelofs, J.G.M., 2003.  
781 Effects of mining activities on heavy metal concentrations in water, sediment, and  
782 macroinvertebrates in different reaches of the Pilcomayo River, South America. *Arch. Environ.*  
783 *Contam. Toxicol.* 44, 314–323.

784 Statzner, B., Beche, L.A., 2010. Can biological invertebrate traits resolve effects of multiple  
785 stressors on running water ecosystems? *Freshwater Biol.* 55(s1), 80–119.

786 Sutherland, A.B., Culp, J.M., Benoy, G.A., 2012. Evaluation of deposited sediment and  
787 macroinvertebrate metrics used to quantify biological response to excessive sedimentation in  
788 agricultural streams. *Environ. Manage.* 50(1), 50–63.

789 Turley, M.D., Bilotta, G.S., Extence, C.A., Brazier, R.E., 2014. Evaluation of a fine sediment  
790 biomonitoring tool across a wide range of temperate rivers and streams. *Freshwater Biol.* 59(11),  
791 2268–2277.

792 Turley, M.D., Bilotta, G.S., Krueger, T., Brazier, R.E., Extence, C.A., 2015. Developing an  
793 improved biomonitoring tool for fine sediment: Combining expert knowledge and empirical  
794 data. *Ecol. Indic.* 54, 82–86.

795 Turley, M.D., Bilotta, G.S., Chadd, R.P., Extence, C.A., Brazier, R.E., Burnside, N.G., Pickwell,  
796 A.G., 2016. A sediment-specific family-level biomonitoring tool to identify the impacts of fine  
797 sediment in temperate rivers and streams. *Ecol. Indic.* 70, 151–165.

798 Usseglio-Polatera, P., Bournaud, M., Richoux, P., Tachet, H., 2000. Biological and ecological traits  
799 of benthic freshwater macroinvertebrates: relationships and definition of groups with similar  
800 traits. *Freshwater Biol.* 43(2), 175–205.

801 Vannote, R.L., Minshall, G.W., Cummins, K.W., Sedell, J.R., Cushing, C.E., 1980. The river  
802 continuum concept. *Can. J. Fish. Aquat. Sci.* 37(1), 130–137.

803 Venables, W.N., Ripley, B.D., 2002. *Modern Applied Statistics with S*, fourth ed. Springer, New  
804 York.

805 Vlek, H.E., Verdonchot, P.F., Nijboer, R.C., 2004. Towards a multimetric index for the assessment  
806 of Dutch streams using benthic macroinvertebrates. *Hydrobiologia* 516, 173–189.

807 Wagenhoff, A., Townsend, C.R., Phillips, N., Matthaei, C.D., 2011. Subsidy-stress and multiple-  
808 stressor effects along gradients of deposited fine sediment and dissolved nutrients in a regional set  
809 of streams and rivers. *Freshwater Biol.*, 56(9), 1916–1936.

810 Wilkes, M.A., McKenzie, M., Murphy, J.F., Chadd, R.P., 2017. Assessing the mechanistic basis for  
811 fine sediment biomonitoring: inconsistencies among the literature, traits and indices. *River Res.*  
812 *Appl.* DOI: 10.1002/rra.3139.

813 Wohl, E., 2006. Human impacts to mountain streams. *Geomorphology* 79(3), 217–248.

814 Wood, P.J., Armitage, P.D., 1997. Biological effects of fine sediment in the lotic  
815 environment. *Environ. Manage.* 21(2), 203–217.

816 Wood, P.J., Armitage, P.D., 1999. Sediment deposition in a small lowland stream—management  
817 implications. *River Res. Appl.* 15, 199–210.

818 Zweig, L.D., Rabeni, C.F., 2001. Biomonitoring for deposited sediment using benthic invertebrates:  
819 a test on 4 Missouri streams. *J. N. Am. Benthol. Soc.* 20(4), 643–657.